

## Do toxic baits containing sodium fluoroacetate (1080) affect fish and invertebrate communities when they fall into streams?

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**Abstract** Large-scale control of Australian possums throughout New Zealand uses toxic cereal baits containing 1080 (sodium fluoroacetate). These baits are often aerially applied over rough terrain where ground application is impractical. Many small streams flow in these areas, so 1080 baits can potentially fall into them. The ecological effect of 1080 leaching from baits was assessed on freshwater fish and invertebrate communities in four streams using a BACI experimental design. Four sites were selected in each stream: 10 m and 100 m below and 10 m and 100 m above where 1080 baits were placed. All sites were monitored 4 days and 1 day before and after baits were added, respectively. Separate experiments were conducted to assess impacts of 1080 on native fish and invertebrate communities. Baits were added to each stream to achieve bait densities 10 × greater than found after normal control operations. Three species of native fish, longfin eels (*Anguilla dieffenbachii*), koaro (*Galaxias brevipinnis*), and upland bullies (*Gobiomorphus breviceps*) were placed into separate cages at each site in each stream, and mortality monitored during the experiment. Analysis of water samples collected during the fish experiment showed that 1080 was detected only for 12 h, and at low concentrations (c. 0.2 µg litre<sup>-1</sup>), despite the large number of baits placed in each stream. No fish died after addition of 1080

baits, suggesting that all three species were tolerant to dissolved 1080 at concentrations observed in this study. Invertebrate communities were quantified by sampling 10 replicate rocks at each site. Caddisflies (*Helicopsyche*, *Pycnocentroides*, and *Pycnocentria*), orthoclad midges, and the mayfly *Deleatidium* dominated the community. 1080 had no detectable effects on the invertebrate community. These results suggest that 1080 leaching from submerged baits in small streams has no demonstrable biological impacts. Based on this finding, the need to maintain buffer zones around large waterways as some councils require is questioned.

**Keywords** invertebrates; native freshwater fish; sodium fluoroacetate; 1080; toxicity testing; BACI design

### INTRODUCTION

The introduced Australian brushtail possum (*Trichosurus vulpecula*) is common throughout New Zealand, where it jeopardises natural ecosystems and acts as a vector for bovine tuberculosis, threatening the highly valuable dairy and cattle industry (Green 2003). As part of extensive nationwide control operations, the Animal Health Board (AHB) and Department of Conservation (DOC) use cereal baits containing 0.15% sodium fluoroacetate (compound 1080) as a poison to reduce possum densities in areas where they pose risks to either native ecosystems or nearby cattle. Baits are made of cereal, dyed green to reduce their attractiveness to birds, and flavoured with cinnamon to increase their palatability to possums. Two types of baits are commonly produced which differ in their degree of weathering: the more weather resistant Wanganui No. 7 and less resistant RS5 baits. Three bait sizes of each type (c. 2 g, 6 g, and 12 g) are also produced. Although ground application of 1080 baits is preferred, this is not practical in densely forested mountainous areas. Under such conditions, helicopters aerially apply 1080 baits along clearly defined flight paths over

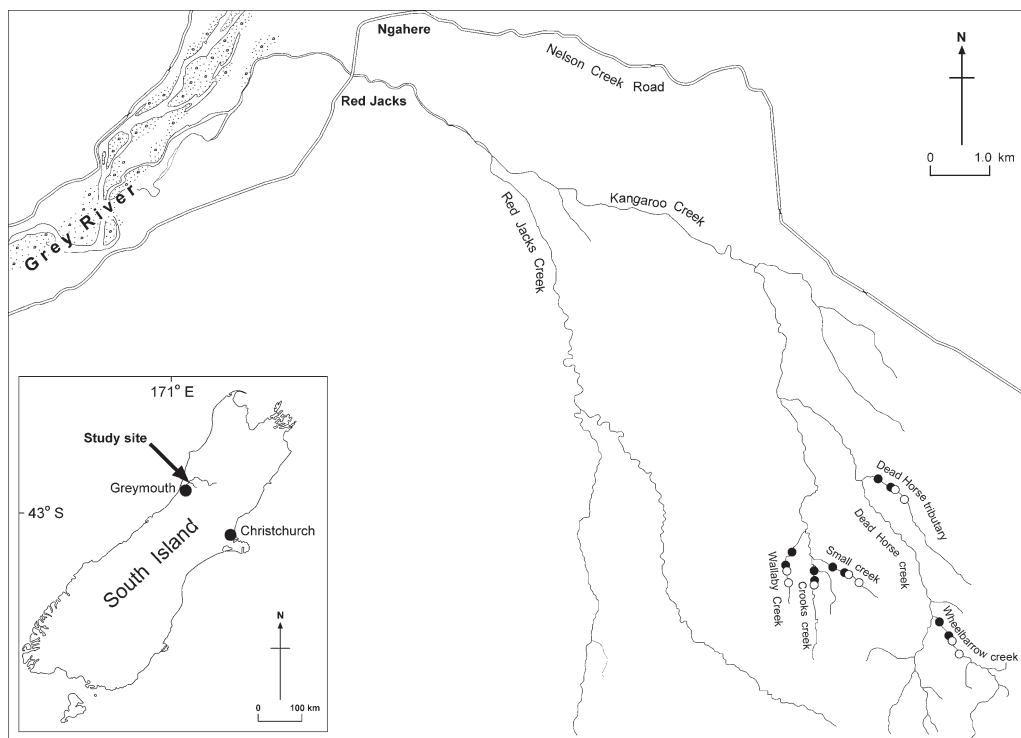
prescribed operational areas. Aerial application is more commonly used in New Zealand than other countries, which generally mostly use ground-baiting operations for pest control to minimise adverse effects on their indigenous mammal populations. Lack of indigenous mammals in New Zealand means that such concerns are unwarranted. New Zealand aerial operations are controlled by resource consents, many of which require that rivers >3 m have buffers around them to minimise accidental contamination of streams by 1080 baits. However, smaller streams have no such buffers, and 1080 baits often fall into these (Suren 2006).

Fish living in streams may be exposed to 1080 leaching from submerged baits, with possible adverse impacts. Previous studies have shown that a wide variety of fish are tolerant to dissolved 1080, even at high concentrations. King & Penfound (1946) found that bass (presumably a species of *Micropterus*: Centrarchidae) survived without signs of toxicity in water containing  $370 \mu\text{g litre}^{-1}$  of 1080; Batcheler (1978) showed that rainbow trout (*Oncorhynchus mykiss*: Salmonidae) survived in  $580 \mu\text{g litre}^{-1}$  of 1080 for 24 h, and Fagerstone et al. (1994) found that bluegill sunfish (*Lepomis macrochirus*: Centrarchidae) showed no signs of mortality over 96 h to  $970 \mu\text{g litre}^{-1}$  of 1080. 1080 is thus regarded as being either non-toxic, or only slightly toxic to fish, depending on the species (Fagerstone et al. 1994). Despite information on toxicity of 1080 to the above three fish, potential effects on New Zealand native fish are unknown. Although 1080 concentrations that fish have been experimentally exposed to have ranged from 90 to  $240 \times$  the maximum concentration detected in water monitoring programmes ( $4 \mu\text{g litre}^{-1}$ ) following aerial application of 1080 within New Zealand (Eason 2002; Green 2003), potential effects of 1080 on native fish need to be assessed, especially given the contentious nature of the use of 1080 within New Zealand (Livingston 1994; Suren 2006).

Baits falling into streams may affect aquatic invertebrates, given the documented toxicity of 1080 to terrestrial invertebrates (Eason et al. 1993; Eisler 1995; Spurr & Drew 1999). Detritivores such as stoneflies (e.g., species of *Acroperla*, *Austroperla*, *Zelandoperla*), caddisflies (*Zelandopsyche*), and some tipulids may be especially vulnerable, as they could potentially consume fragments of 1080 bait. However, it is highly unlikely that these animals would consume whole baits, which are considerably larger than individual animals. By the time baits fragment, most, or all, of the 1080 would have leached from them (Suren 2006), so the chance of

invertebrates suffering mortality from consuming fragments is negligible. Nevertheless, freshwater invertebrates may be affected by 1080 leaching from submerged baits, especially in small streams. Toxicity tests performed by the United States Environmental Protection Agency have shown that the non-observable effect concentration (NOEC) of 1080 for the small freshwater invertebrate *Daphnia magna* was  $130 \text{ mg litre}^{-1}$  (Fagerstone et al. 1994). At this high concentration, the effect of 1080 was regarded as being "practically non-toxic" to *Daphnia*. Another study of 1080 toxicity to fourth instar mosquito larvae (*Anopheles quadrimaculatus*) showed that 1080 was among the most toxic 3% of 6000 organic compounds screened, with 65% mortality at concentrations of  $100 \mu\text{g litre}^{-1}$  over a 48 h period (Deonier et al. 1946). These two concentrations are about  $30 \times$  higher than the maximum concentration of 1080 detected in streams in New Zealand (Green 2003), so it seems unlikely that 1080 baits falling into streams would have a toxic effect on New Zealand invertebrates. Despite this assertion, no studies have quantified the toxicity of 1080 to any New Zealand freshwater invertebrates, nor assessed the effects of submerged baits on freshwater invertebrates.

Despite the widespread use of 1080, and its potential to adversely impact aquatic organisms, only two published reports have attempted to quantify such impacts in New Zealand (Taranaki Regional Council 1993, 1994). Both studies used qualitative kick samples to characterise invertebrate communities in streams flowing through forested areas where 1080 had been applied aerially. Invertebrate samples were collected from streams before and after 1080 application, and a number of biotic indices calculated to describe the invertebrate communities. No demonstrable effects of the aerial 1080 application were observed (Taranaki Regional Council 1993, 1994), suggesting that stream invertebrates were not sensitive to 1080. Biological samples for these studies were collected before, and between 32 and 42 days after the aerial application of 1080. This sampling protocol may thus have missed any short-term reductions in invertebrate density arising as a result of any toxic effect of 1080. Moreover, these two studies made the implicit assumption that they adequately sampled areas where 1080 contamination had occurred. This was, however, not controlled for, and recent work (Suren 2006) has shown baits do not necessarily land in streams flowing in areas where 1080 is aerially applied. It could be argued that although these monitoring studies demonstrated no adverse impacts of 1080 operations at a catchment



**Fig. 1** Site map showing the five small streams selected for study. Three of the streams were used for both the fish and invertebrate study, whereas Wallaby Creek was used for the fish study only, and Small creek for the invertebrate study only. Within each stream, there were two sites above locations where 1080 (sodium fluoroacetate) baits were added (open circles), and two sites below (closed circles). Note that Crooks creek only had one control site for the fish study as cages had been stolen from the upper control site. For the invertebrate study, two control sites were used at this stream.

scale, they did not adequately quantify the effects of 1080 bait in individual streams. Finally, neither study specifically examined the effects of 1080 contamination on fish communities.

Continued public concern exists about the environmental fate of 1080 (e.g., Laugesen & Hubbard 2002; Speedy 2003), especially in water. The present study was designed as a field-based experiment at the scale of individual catchments to examine the effect of submerged 1080 baits on two trophic levels within streams: freshwater fish and invertebrates.

## MATERIALS AND METHODS

### Study sites

Five headwater tributary streams of Dead Horse and Wallaby creeks in the Mawhera State Forest in the Grey Valley, on the west coast of the South Island,

were selected for the study (Fig. 1). All streams were relatively remote from human habitation, and small (<3 m wide), so would not have buffers around them in the course of normal 1080 operations. They thus represented streams likely to become contaminated by 1080. Dead Horse tributary, Wallaby Creek and Small creek flowed through catchments dominated by plantation pines (*Pinus radiata*), whereas Crooks creek and Wheelbarrow creek flowed from upper catchments of native bush (mainly podocarp forest) into lower catchments of pine. The immediate riparian vegetation of all streams was a mixture of pines, native scrub, and grasses. Boulders, cobbles and gravels dominated the streambed material at all sites. Velocity was measured at 20 vertical cross-sections across transects in each stream ( $0.4 \times \text{depth}$ ) using a small Ott meter, at the beginning and end of the experiment, and immediately before 1080 baits were added. Transects were located within 20 m of where 1080 baits were added. Discharge

in each stream was calculated (to within  $\pm 6.9\%$ ) using the TDGAUGE (2005) programme. Stream pH (to  $\pm 0.01$  units), and conductivity (to within  $\pm 4.4\%$ ) were measured on each of the four sampling periods using a TPS WP81 pH/conductivity meter, and temperature and dissolved oxygen (to within  $\pm 0.02$  ppm) were measured using a TPS WP82 oxygen sensor. These measurements were taken at the same localities as each gauging.

We planned to conduct the fish and invertebrate experiments simultaneously in March 2004. However, high rainfall and river flows in the region during February 2004 resulted in low invertebrate densities that would have been insufficient to detect any potential effects of 1080. The invertebrate component of the study was thus delayed until mid May when invertebrate densities had recovered. Wallaby Creek was not used for the invertebrate experiment, as large amounts of filamentous green algae covered the streambed at that time, reducing habitat conditions for invertebrates. We subsequently chose a new stream (Small creek) that was near Crooks creek for this experiment (Fig. 1). The fish experiments were conducted in a lower 200 m section of each stream, while the invertebrate experiments were conducted in a 200 m section upstream of where 1080 baits were placed for the fish experiment, to ensure independence of each experiment. Within each stream, two control sites were selected c. 100 m and 10 m upstream from where 1080 baits were placed (sites 1 and 2), and two impact sites c. 10 m and 100 m downstream from these baits (sites 3 and 4).

### Field methods—fish study

The impact of 1080 leaching from baits was examined on three fish species that are common on the west coast: longfin eels (*Anguilla dieffenbachii* Gray (Anguillidae)), koaro (*Galaxias brevipinnis* Günther (Galaxiidae)) and upland bullies (*Gobiomorphus breviceps* (Stokell) (Gobiidae)). These fish were caught by electric fishing from a number of small streams in the Grey Valley. All fish were transported to experimental streams within 12 h of collection in plastic buckets equipped with battery-operated air pumps to ensure constant aeration.

Fish cages were constructed from 250-litre plastic barrels cut in half longitudinally and fitted with 2 mm mesh netting on their upstream and downstream ends. A 2 mm mesh lid on the top was sealed with Velcro. Up to 10 large cobbles (mean largest diam. 140 mm) were added to each cage to provide shelter. Four PVC tubes (24 mm diam. and 200 mm length) were also added to cages holding eels, to provide

additional shelter. Three cages (one cage per species) were deployed in relatively deep runs or pools (up to 40 cm deep) at each of the four sites, which were excavated until cages were submerged. Each cage was anchored to the streambed, and large cobbles placed on top to weigh them down and ensure their mesh lids remained closed. Once deployed, eight fish of each species were placed into individual cages at all sites. Aquatic insects that naturally drifted into the cages provided the fish with food. Extra invertebrates (obtained from the streams) were added to the cages every 4 days to minimise any food limitation. All cages at site 1 were stolen from Crooks creek before the experiment started, leaving control cages at site 2 only.

Fish were added to all cages on 14 March 2004 and mortality recorded 1 and 4 days later (15 and 18 March 2004). On each occasion, all cobbles and PVC pipes (in the eel cages) were removed from each cage, which was then emptied into a large collecting bucket. All fish were counted and dead fish recorded. The cobbles and PVC pipes were replaced into each cage, which was then partially submerged. The fish were subsequently poured from the collecting bucket back into each cage, the Velcro-sealed mesh lid was resealed, and the cage repositioned under water.

After the second observation of fish survival on 18 March 2004, 1080 baits (Wanganui No. 7, mean weight = 6.5 g) were added to each stream. We wanted to expose the two impact sites, especially site 3 (10 m below the baits) to a high level of 1080 leaching from baits. Suren (2006) quantified the number of baits falling into streams flowing through areas where 1080 had been aerially applied during possum control operations, and found that the highest number of baits per 10 m of stream was eight. We thus added 80 baits ( $10 \times$  the maximum number) into the stream with the highest discharge (Wheelbarrow creek). The other three creeks had lower discharges, so the number of baits added to these was calculated according to the ratio of their discharge to that of Wheelbarrow creek (Table 1). Baits were counted into three sets of nylon mesh bags (mesh size 10 mm) that were submerged and anchored to the streambed at three evenly spaced locations across each stream to ensure complete mixing of leached 1080.

Observations of fish survival at all four sites were made 1 and 4 days after addition of 1080 (19 and 22 March 2004). On the last occasion, all fish were anaesthetised and their lengths measured to the nearest mm. Individual water samples (1 litre) were collected from all sites concurrently

with observations of fish on all four dates from the centre of each stream. Water samples were also collected from the two impact sites 2, 4, and 8 h after addition of baits, reflecting the rate of 1080 leaching from baits (Suren 2006). All water samples were frozen ( $-18^{\circ}\text{C}$  within 6 h of collection) and analysed for dissolved 1080 concentration using gas chromatography, method TLM 005 (Landcare Research Ltd, see Lyver et al. 2005). The limit of detection was  $0.001\ \mu\text{g ml}^{-1}$  (0.1 ppb).

### Field methods—*invertebrate study*

The effect of 1080 leaching from baits was assessed on invertebrate communities in each stream in May 2004. Invertebrates were collected using the “rock rolling” technique (see Death & Winterbourn 1994; Matthei et al. 2000). Here, a net (mesh size  $300\ \mu\text{m}$ ) was placed immediately below individual rocks, which were lifted quickly into the net. Animals on each rock were either immediately dislodged and collected in the net, or removed from the rock by scrubbing. This technique was used as the boulder and cobble-dominated streambed precluded use of traditional quantitative techniques (e.g., Stark et al. 2001). Ten replicate rocks were randomly sampled in riffles at each site, and rock dimensions (length  $\times$  width  $\times$  height) measured after sampling to calculate their surface areas (Biggs & Kilroy 2000) to estimate invertebrate densities. A power analysis of invertebrate data collected previously from individual rocks in Dead Horse creek showed that 10 replicate rocks were sufficient to detect a 20% reduction in total invertebrate density, and densities of common taxa, with an 80% certainty (A. Suren unpubl. data). As invertebrate densities in small streams can vary monthly by up to 30%

of the long-term (18-month) average (Suren 1991, unpubl. data), the ability of the present analysis to detect a smaller density reduction (20%) as a result of 1080 over an 8-day period was considered sufficiently robust.

Invertebrates were first collected from all sites on 20 May 2004, and 4 days later (24 May 2004), after which 1080 baits were added to the streams in the same manner as for the fish experiment. Discharge in the streams was less than for the fish experiment, so fewer baits were used (Table 1). It was, however, decided not to reduce the number of baits added to each stream in direct proportion to the lower stream discharges, as this may have been perceived as being too conservative. Instead, enough baits were added so that predicted 1080 concentrations were almost three times higher than those used in the fish trials (Table 1). The invertebrate experiment therefore represented an extreme scenario following aerial application of 1080. Invertebrate samples were collected 1 day (25 May 2004) and 4 days (28 May 2004) after addition of 1080. Water samples were also collected from both impact sites in each stream 1 and 4 days after addition of baits, and processed in a similar manner as samples collected in the fish experiment.

All invertebrate samples were preserved in the field using 50% iso-propanol, and returned to the laboratory. Samples were processed using a modification of Protocol P3 (Stark et al. 2001), whereby all samples were placed in a small Bogorov tray (see Winterbourn & Gregson 1989) and scanned under a stereo-microscope for identification and enumeration of all invertebrates. This method allowed small animals such as small chironomids, nematodes, copepods, and ostracods

**Table 1** Estimated 1080 (sodium fluoroacetate) concentrations in each of the experimental streams, based on stream discharge when baits were added, the number of baits added, and assuming complete dissolution in 8 h (based on Suren 2006).

Stream	Discharge when baits added ( $\text{litres s}^{-1}$ )	No. of baits	1080 weight added (mg)	Estimated concentration ( $\mu\text{g litre}^{-1}$ )
<b>Fish study</b>				
Crooks creek	56	30	288	0.18
Dead Horse tributary	86	50	480	0.19
Wallaby Creek	51	30	288	0.20
Wheelbarrow creek	105	80	768	0.25
<b>Invertebrate study</b>				
Crooks creek	16	20	192	0.42
Dead Horse tributary	14	20	192	0.48
Small creek	6	15	144	0.83
Wheelbarrow creek	17	25	240	0.49

to be counted accurately. All mollusca and aquatic insects were identified to genera using standard keys (e.g., Winterbourn 1973, 1989), except for the Chironomidae, which were identified to subfamily. Other taxa were identified to either subclass (e.g., Crustacea) or family (Oligochaeta).

### Statistical analysis

Two-way ANOVA (SPSS 2000) was used to determine differences in discharge, measured water quality parameters and fish length between streams, or between control and impact sites. Three-way ANOVA was used to test for differences in stone sizes between streams, control and impact sites, or before and after 1080 baits were added. Untransformed data was used for all these analyses except for discharge data, which was  $\log_{10}$ -transformed before analysis.

A number of metrics were calculated from the invertebrate data, including total invertebrate density, taxonomic richness, densities of the four most common and widespread taxa, the number of Ephemeroptera, Plecoptera and Trichoptera (EPT) taxa, and the % of EPT taxa. The Macroinvertebrate Community Index (MCI) score, and its quantitative variant (QMCI: Stark 1985, 1993) were also calculated. Although the latter two metrics were originally derived to assess organic pollution in stony streams (Stark 1985, 1993), they are appropriate for detecting changes to invertebrate communities as a result of 1080, as changes to community structure would manifest themselves as a change to calculated MCI or QMCI scores.

Invertebrate data were analysed by ANOVA using the GENSTAT package (GENSTAT 2001). All data were examined for normality and  $\log_{10}$ -transformed where necessary. The study was based on a Before-After-Control-Impact (BACI) design, used to detect changes in biological communities as a result of human activities (Underwood 1991), where samples collected above 1080 baits were considered controls and samples collected below baits were impact sites. Although traditional BACI tests have two main experimental effects (usually a time and site effect), our experiment was not straightforward. First, the two sites below the 1080 baits represented a dilution gradient, where the lowermost site could have been exposed to lower 1080 concentrations than the upper site as a result of groundwater inputs and possible breakdown of 1080. Secondly, invertebrate densities may have recovered in samples collected 4 days after bait addition as a result of upstream immigration, whereas such immigration would have been much

less 1 day after the introduction of 1080. Because of these complications, extra terms were included in the ANOVA model: (1) to separate potential dilution effects, the "site" term was partitioned into a "Control" versus "Impact" main effect, a "100 m" versus "10 m" main effect, and a spatial interaction between "Control versus Impact"  $\times$  "100 m versus 10 m"; (2) to separate any recovery in invertebrate communities as a result of immigration, the "time" term was partitioned into a "Before" versus "After" main effect as well as a "4 day" versus "1 day" main effect, and a temporal interaction between "Before versus After"  $\times$  "4 day versus 1 day".

The ANOVA model had three main effects (site, time, stream), as well as all interaction effects. We were particularly interested in interaction effects that suggested that sites below the 1080 baits were different to those above. The relevant interaction terms that indicated this specific difference were: (1) "Control versus Impact"  $\times$  "Before versus After" effect; (2) the spatial interaction term  $\times$  "Before versus After"; and (3) the temporal interaction term  $\times$  "Control versus Impact". These interaction terms were examined for the combined data set, and for interactions with the "stream" main effect.

The effect of 1080 on invertebrate community composition in the streams was also assessed using a multivariate approach. First, a matrix of Bray-Curtis similarity coefficients was constructed using abundance data from all possible pairs of invertebrate samples (Digby & Kempton 1987). Bray-Curtis coefficients range from 0 (no common taxa between the two sites) to 1 (all species common to both sites, with the same relative abundances). Differences in Bray-Curtis scores of samples collected at the different times and sites were analysed using PERMANOVA (Anderson 2001) to assess whether placement of 1080 baits changed the downstream invertebrate communities relative to those upstream. The PERMANOVA procedure was set to run 5000 different permutations to test whether observed differences could have arisen owing to chance, or whether they were indicative of an actual effect. Significant levels for all tests were set at  $P < 0.05$ .

## RESULTS

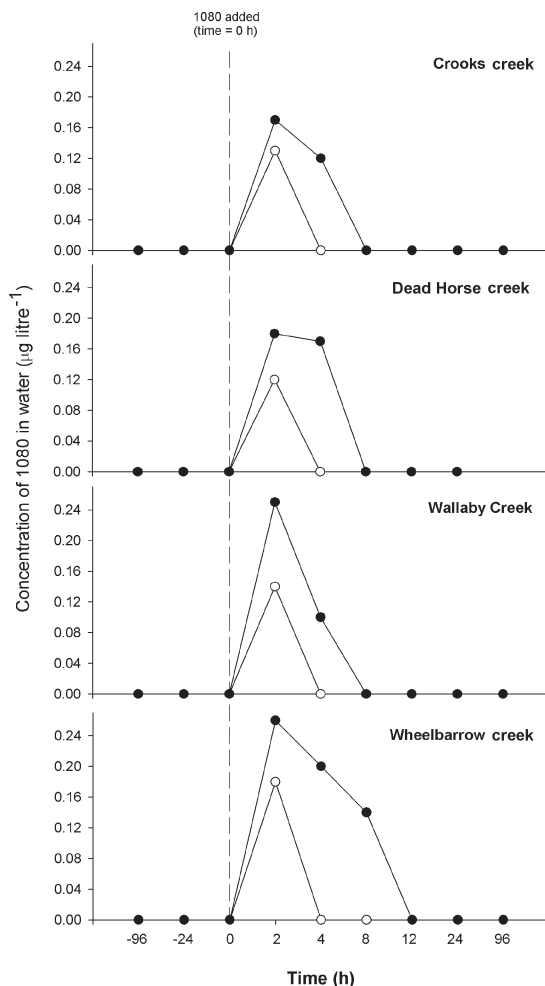
### Effects on native fish

Discharge differed significantly between streams ( $F_{(3,8)} = 4.67$ ,  $P < 0.05$ ), and was highest in Dead Horse tributary and lowest in Wallaby Creek (Table 2). Discharge decreased slightly during the study,

but this was not significant ( $F_{(1,8)} = 0.70$ ,  $P > 0.05$ ). Stream pH ranged from 4.1 to 6.4, and was significantly lower in Wheelbarrow creek and Dead Horse tributary ( $F_{(3,8)} = 54.94$ ,  $P < 0.001$ ), and higher in Wallaby Creek (Table 2). Stream conductivity was low, but significantly higher in Wallaby Creek and Crooks creek (Table 2) than the other streams ( $F_{(3,8)} = 38.97$ ,  $P < 0.001$ ). Dissolved O<sub>2</sub> was relatively low (c. 75% saturated) and similar in all streams ( $F_{(3,8)} = 0.35$ ,  $P > 0.05$ ), whereas temperature was highest in Wallaby Creek ( $F_{(3,8)} = 35.85$ ,  $P < 0.001$ , Table 2).

1080 was detected in water samples collected during the fish experiment from impact sites only for a short duration (up to 8 h) after addition of baits (Fig. 2). 1080 concentrations were higher at sites 10 m below the baits (mean  $\pm$  1SD:  $0.176 \pm 0.054 \mu\text{g litre}^{-1}$ ) than sites 100 m below this point ( $0.063 \pm 0.076 \mu\text{g litre}^{-1}$ ). No 1080 was found in any samples collected from impact sites after 12 h, or from any control sites.

There was no significant difference in bully length ( $54 \pm 9$  mm) in any of the streams ( $F_{(3,93)} = 0.714$ ,  $P > 0.05$ ), or in cages placed in control or impact sites ( $F_{(1,93)} = 0.6545$ ,  $P > 0.05$ ). Mean lengths of longfin eels differed significantly between streams



**Fig. 2** Concentration of 1080 (sodium fluoroacetate) in water samples ( $\mu\text{g litre}^{-1}$ ) collected from the four streams during the fish experiment. Samples were collected from sites 10 m below (closed symbols) and 100 m below (open symbols) 1080 baits on three occasions before baits were added, and six occasions after. Note that some symbols from the 10 m site obscure those from the 100 m site.

**Table 2** Summary of the physical and water-quality conditions of the five study streams. Stream discharge was measured at the site where 1080 (sodium fluoroacetate) baits were added, at the beginning, immediately before 1080 baits were added, and at the end of each experiment. Other parameters were measured on all sampling occasions, and are averages ( $\pm$  1 SD,  $n = 4$ ). All stream names except Wallaby Creek are unofficial.

Stream	Range of discharge (litres $\text{s}^{-1}$ )	pH	Conductivity ( $\mu\text{S cm}^{-1}$ )	Dissolved O <sub>2</sub> (mg litre <sup>-1</sup> )	Temperature ( $^{\circ}\text{C}$ )
<b>Fish study</b>					
Crooks creek	56–125	$5.6 \pm 0.1$	$31 \pm 2$	$8.1 \pm 0.4$	$12.5 \pm 0.3$
Dead Horse tributary	86–305	$4.7 \pm 0.3$	$25 \pm 1$	$7.9 \pm 0.3$	$12.5 \pm 0.6$
Wallaby Creek	51–127	$6.2 \pm 0.2$	$38 \pm 3$	$8.1 \pm 0.6$	$14.7 \pm 0.9$
Wheelbarrow creek	78–238	$4.5 \pm 0.3$	$26 \pm 2$	$7.9 \pm 0.2$	$12.5 \pm 0.5$
<b>Invertebrate study</b>					
Crooks creek	15–17	$5.9 \pm 0.3$	$34 \pm 6$	$11.2 \pm 0.1$	$7.6 \pm 0.3$
Dead Horse tributary	12–15	$5.8 \pm 0.2$	$32 \pm 1$	$11.9 \pm 0.1$	$6.5 \pm 0.7$
Small creek	6–7	$6.4 \pm 0.3$	$53 \pm 2$	$10.5 \pm 0.3$	$8.7 \pm 0.3$
Wheelbarrow creek	16–18	$5.6 \pm 0.2$	$34 \pm 1$	$10.8 \pm 0.2$	$7.4 \pm 0.7$

( $F_{(3, 102)} = 15.76, P < 0.001$ ), and were longest at Crooks creek ( $339 \pm 17$  mm) and the same length at the other streams ( $256 \pm 77$  mm). Longfin eel lengths at impact sites below the 1080 baits ( $256 \pm 85$  mm) were also less than those at control sites ( $294 \pm 79$  mm;  $F_{(1, 102)} = 16.37, P < 0.001$ ). There was no significant difference in koaro lengths ( $86 \pm 21$  mm) in any of the streams ( $F_{(3, 93)} = 1.258, P > 0.05$ ), or in control or impact sites ( $F_{(1, 93)} = 0.005, P > 0.05$ ).

No fish mortality was observed in any of the cages except at Wheelbarrow creek, where 15 out of 32 upland bullies and one koaro had died between the first and second sampling occasions (Table 3). This mortality appeared to be owing to heavy rainfall between these time periods, which overturned some of the cages and partially filled them with fine sand. Although some fish (notably eels) escaped from cages (Table 3), sufficient fish remained in cages at all impact sites to monitor mortality arising as a result of 1080 exposure. No fish died in any of the streams after 1080 baits were added (Table 3).

### Effects on invertebrate communities

Discharge was much lower for the invertebrate experiment (Table 2), and was significantly lower in Small creek ( $F_{(3,8)} = 22.16, P < 0.001$ ) than in the other three streams. Stream pH was significantly lower in Wheelbarrow creek and Dead Horse tributary ( $F_{(3,8)} = 6.87, P < 0.01$ ), whereas conductivity was significantly higher in Small creek ( $F_{(3,8)} = 62.59, P < 0.001$ ).

Dissolved  $O_2$  was higher in the four streams than for the fish experiment (Table 2) and was significantly higher in Dead Horse tributary ( $F_{(3,8)} = 28.84, P < 0.001$ ), whereas water temperature was cooler, and significantly lower in Dead Horse tributary ( $F_{(3,8)} = 16.65, P < 0.01$ ). Rock sizes ( $162 \pm 37$  mm  $\times$   $119 \pm 26$  mm  $\times$   $70 \pm 21$  mm) were similar between streams ( $F_{(3, 624)} = 2.58, P > 0.05$ ), and between control and impact sites ( $F_{(1, 624)} = 1.11, P > 0.05$ ). There was also no significant difference in rock sizes collected before or after placement of 1080 baits ( $F_{(1,624)} = 0.67, P > 0.05$ ).

**Table 3** Fish loss observed during the experiment, based on eight fish of each species (upland bullies (*Gobiomorphus breviceps*) (b), eel (*Anguilla dieffenbachii*) (e), and koaro (*Galaxias brevipinnis*) (k) originally added to individual cages. Fish survival was monitored 1 and 4 days before 1080 (sodium fluoroacetate) baits were added (−4 and −1 d), and 1 and 4 days after (+1 and +4 d), at locations above and below bait placement. Cages placed below 1080 baits are shown in italics. Reductions in numbers owing to mortality indicated in bold, other reductions in numbers were owing to fish escaping from cages. Cages at site 1 in Crooks creek were stolen, so no data were collected (nd).

Stream	Time (days)	Above 1080 baits		Below 1080 baits	
		Site 1–100 m (b, e, k)	Site 2–10 m (b, e, k)	Site 3–10 m (b, e, k)	Site 4–100 m (b, e, k)
Crooks creek	−4	nd	0, 0, 0	0, 0, 0	0, 0, 0
	−1	nd	0, 0, 0	0, 0, 0	1, 0, 1
	+1	nd	0, 0, 0	<i>0, 0, 0</i>	<i>1, 0, 1</i>
	+4	nd	0, 0, 0	<i>0, 0, 0</i>	<i>1, 0, 1</i>
Dead Horse tributary	−4	0, 0, 0	0, 0, 0	0, 0, 0	0, 0, 0
	−1	0, 0, 0	0, 0, 0	0, 0, 1	0, 4, 0
	+1	0, 0, 0	1, 0, 0	<i>0, 0, 1</i>	<i>0, 4, 0</i>
	+4	0, 0, 0	1, 0, 0	<i>1, 0, 1</i>	<i>0, 4, 0</i>
Wheelbarrow creek	−4	0, 0, 0	0, 0, 0	0, 0, 0	0, 0, 0
	−1	<b>8, 2, 0</b>	<b>3, 0, 1</b>	<b>2, 0, 0</b>	<b>2, 0, 0</b>
	+1	<b>8, 2, 0</b>	<b>3, 0, 1</b>	<b>2, 0, 0</b>	<b>2, 0, 0</b>
	+4	<b>8, 2, 0</b>	<b>3, 0, 1</b>	<b>2, 0, 0</b>	<b>2, 0, 0</b>
Wallaby Creek	−4	0, 0, 0	0, 0, 0	0, 0, 0	0, 0, 0
	−1	0, 0, 0	0, 0, 1	0, 0, 0	0, 3, 0
	+1	0, 0, 0	0, 0, 1	<i>0, 0, 0</i>	<i>0, 3, 0</i>
	+4	0, 0, 0	0, 0, 1	<i>0, 0, 0</i>	<i>0, 3, 0</i>

A total of 72 taxa and 87889 individuals were collected from the rocks during the study, most common of which was the caddisfly *Helicopsyche*, contributing up to 35% to total density, followed by orthoclad midges (18%), the leptophlebid mayfly *Deleatidium* (10%), and the caddisflies *Pycnocentroides* and *Pycnocentria* (7% each). The stonefly *Zelandoperla*, and the crane fly *Aphrophilia* were also relatively common, contributing 3% to total density. Only 12 taxa were considered common (densities >1%), 22 taxa were considered occasional (densities between 0.1 and 1%), and 38 taxa were considered uncommon (densities <0.1%). The most widespread taxa were *Helicopsyche*, *Deleatidium*, orthoclad midges, and *Zelandoperla*, which were collected at all sites. Small creek had the highest taxonomic richness (65 taxa) and invertebrate density (mean = 2755 individuals m<sup>-2</sup>), whereas Wheelbarrow

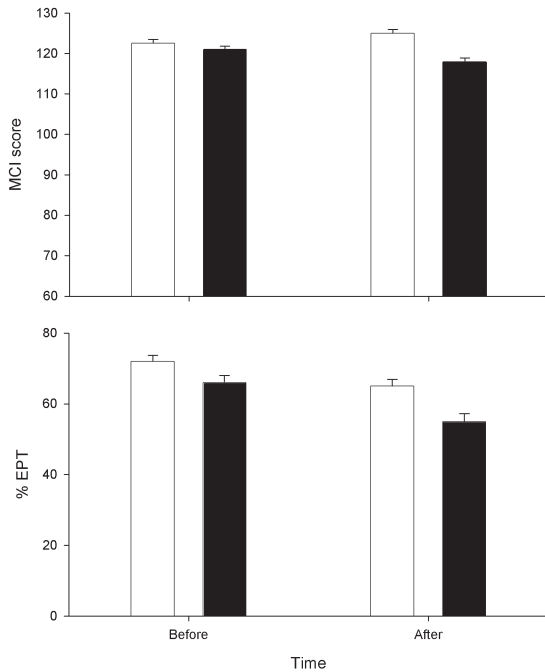
creek had the lowest taxonomic richness (50 taxa) and lowest densities (758 individuals m<sup>-2</sup>).

No 1080 was detected in water samples collected from impact sites 1 day and 4 days after placing between 15 and 25 baits in the streams, despite the very low stream discharge (ranges from 6–17 litres s<sup>-1</sup>, Table 1) which would have minimised dilution effects. Predicted 1080 concentrations in the streams based on an 8 h leaching rate were up to three times higher than those for the fish study (Table 1).

ANOVA showed no large or consistent effects on either community metrics, or densities of common taxa. Few statistically significant interaction terms were observed for the six metrics describing the invertebrate communities that suggested an effect of 1080 (Table 4). The invertebrate fauna at all streams had high MCI scores (121 ± 12) and % EPT (65 ± 25), although both metrics varied widely in the study

**Table 4** Results of ANOVA model for effects of 1080 (sodium fluoroacetate) on selected invertebrate metrics, showing the model interaction effects that were indicative of a potential effect. Values in bold are significant ( $P < 0.05$ ). (MCI, Macroinvertebrate Community Index; QMCI, Quantitative MCI; EPT, Ephemeroptera, Plecoptera, Trichoptera.)

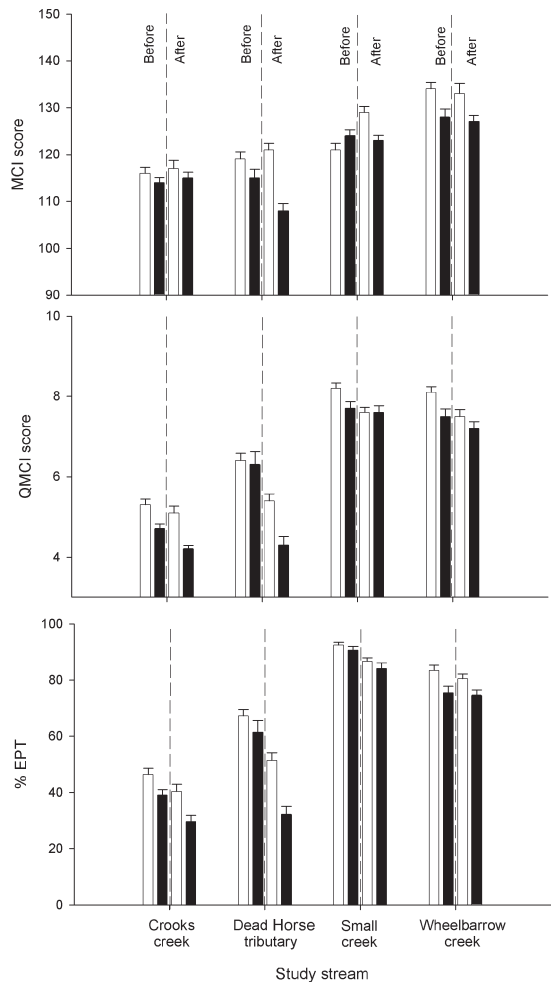
Metric	Model effect	Sums of squares	d.f.	Mean squares	<i>F</i> ratio	<i>P</i>
Density	Control versus Impact × Before versus After	0.082	1	0.082	0.28	0.593
	Spatial interaction × Before versus After	0.611	1	0.611	2.12	0.146
	Temporal interaction × Control versus Impact	0.089	1	0.089	0.31	0.580
	Stream × Control versus Impact × Before versus After	1.664	3	0.555	1.92	0.125
	Error	166.181	576	0.289		
Richness	Control versus Impact × Before versus After	0.017	1	0.017	0.28	0.587
	Spatial interaction × Before versus After	0.004	1	0.004	0.07	0.794
	Temporal interaction × Control versus Impact	0.061	1	0.061	1.02	0.313
	Stream × Control versus Impact × Before versus After	0.353	3	0.118	1.97	0.117
	Error	34.370	576	0.060		
MCI	Control versus Impact × Before versus After	0.204	1	0.204	10.62	<b>0.001</b>
	Spatial interaction × Before versus After	0.030	1	0.030	1.56	0.196
	Temporal interaction × Control versus Impact	0.003	1	0.003	0.16	0.686
	Stream × Control versus Impact × Before versus After	0.120	3	0.040	2.08	<b>0.039</b>
	Error	11.060	576	0.019		
QMCI	Control versus Impact × Before versus After	0.091	1	0.091	0.76	0.383
	Spatial interaction × Before versus After	0.003	1	0.003	0.02	0.884
	Temporal interaction × Control versus Impact	0.023	1	0.023	0.19	0.644
	Stream × Control versus Impact × Before versus After	1.679	3	0.560	4.69	<b>0.003</b>
	Error	68.66	576	0.119		
EPT	Control versus Impact × Before versus After	1.694	1	1.694	2.99	0.084
	Spatial interaction × Before versus After	0.743	1	0.743	1.31	0.252
	Temporal interaction × Control versus Impact	0.018	1	0.018	0.03	0.857
	Stream × Control versus Impact × Before versus After	1.712	3	0.571	1.01	0.388
	Error	325.96	576	0.566		
% EPT	Control versus Impact × Before versus After	615.30	1	615.300	3.89	<b>0.049</b>
	Spatial interaction × Before versus After	155.40	1	155.400	0.98	0.322
	Temporal interaction × Control versus Impact	64.30	1	64.300	0.41	0.524
	Stream × Control versus Impact × Before versus After	1397.80	3	465.933	2.94	<b>0.033</b>
	Error	91182.5	576	158.303		



**Fig. 3** MCI and % EPT ( $\bar{x} \pm 1$  SE,  $n = 160$ ) data for the experimental streams at control (open bars) and impact (closed bars) sites, before and after 1080 (sodium fluoroacetate) bait placement. (MCI, Macroinvertebrate Community Index; EPT, Ephemeroptera, Plecoptera, Trichoptera.)

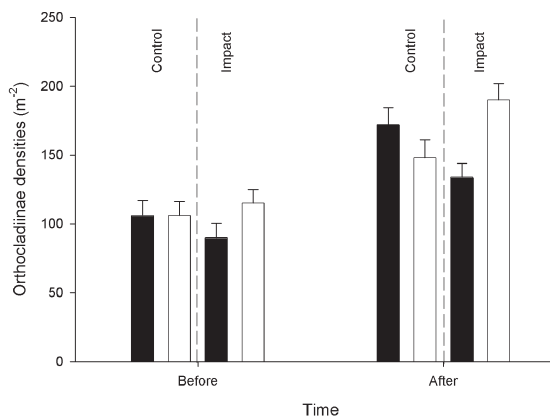
(MCI scores from 88 to 186; % EPT from 10 to 100). These metrics both showed statistically significant “Control versus Impact”  $\times$  “Before versus After” effects (Table 4). Average MCI scores increased slightly (by 2.8) at control sites before and after addition of baits, whereas this metric decreased slightly (by 2) at the impact sites (Fig. 3). Calculated % EPT scores decreased by 11% at impact sites, and decreased by 8% at control sites (Fig. 3).

Patterns in MCI and QMCI scores, and for the % EPT differed between streams (Table 4). MCI scores decreased in Dead Horse tributary at impact sites when compared with control sites (Fig. 4), however the reduction (7) was negligible when compared to the natural range of MCI scores at control sites in this stream (88–150). MCI scores in Small creek increased at control sites, but decreased slightly at impact sites (Fig. 4). MCI scores at Crooks creek remained similar at all sites throughout the study (Fig. 4), whereas scores at Wheelbarrow creek decreased by a similar value in both control and impact sites. QMCI scores decreased more at impact sites than control sites at

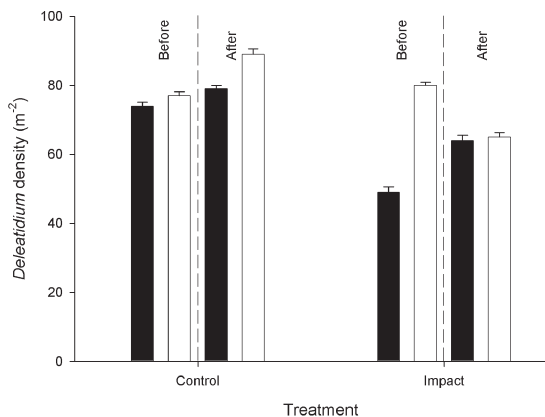


**Fig. 4** MCI and QMCI scores, and % EPT ( $\bar{x} \pm 1$  SE,  $n = 40$ ) data for individual streams at control (open bars) and impact (closed bars) sites, before and after 1080 (sodium fluoroacetate) bait placement. (MCI, Macroinvertebrate Community Index; QMCI, Quantitative MCI; EPT, Ephemeroptera, Plecoptera, Trichoptera.)

Crooks creek and Dead Horse tributary, whereas at the other two sites, QMCI scores decreased more at control sites than at impact sites (Fig. 4). The % EPT at impact sites in Crooks creek and Dead Horse tributary declined more at impact sites (by 10% and 29%, respectively) than at control sites, where it declined by 6% and 15%, respectively. The % EPT declined by a similar value at Wheelbarrow creek (c. 2%) and Small creek (c. 6%) at both control and impact sites (Fig. 4).



**Fig. 5** Densities of orthocladiid midges ( $\bar{x} \pm 1SE, n = 160$ ) at Control versus Impact sites, 10 m (closed symbols) and 100 m downstream (open symbols) from the 1080 (sodium fluoroacetate) baits, before and after 1080 bait placement.



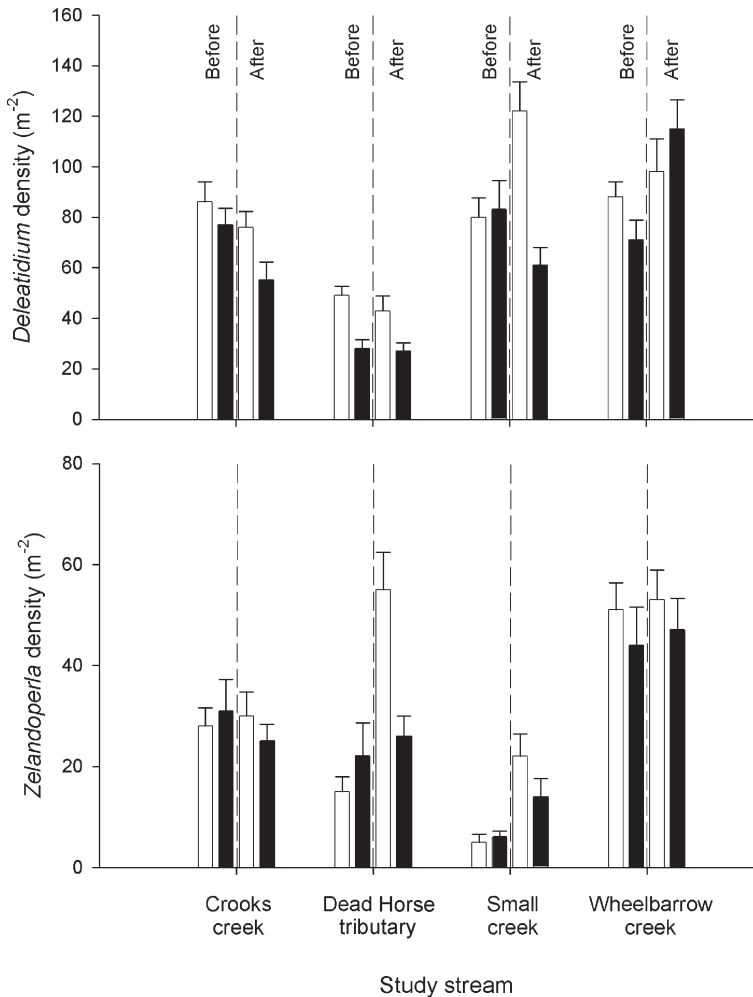
**Fig. 6** Densities of *Deleatidium* nymphs ( $\bar{x} \pm 1SE, n = 160$ ) 1 day (closed bars) and 4 days (open bars) before and after placement of 1080 (sodium fluoroacetate) baits in streams, at Control versus Impact sites.

**Table 5** Results of ANOVA model for effects of 1080 (sodium fluoroacetate) on densities of the four most common and widespread taxa found in the study streams, showing the model interaction effects that were indicative of a potential effect. Values in bold are significant ( $P < 0.05$ ).

Taxa	Model effect	Sums of squares	d.f.	Mean squares	F ratio	P
Deleatidium	Control versus Impact × Before versus After	0.071	1	0.071	0.16	0.685
	Spatial interaction × Before versus After	0.602	1	0.602	1.40	0.237
	Temporal interaction × Control versus Impact	1.823	1	1.823	4.24	<b>0.040</b>
	Stream × Control versus Impact × Before versus After	7.667	3	2.556	5.94	<b>&lt;0.001</b>
	Error	247.709	576	0.430		
<i>Helicopsyche</i>	Control versus Impact × Before versus After	0.486	1	0.486	0.526	0.469
	Spatial interaction × Before versus After	2.597	1	2.597	2.808	0.094
	Temporal interaction × Control versus Impact	0.266	1	0.266	0.288	0.592
	Stream × Control versus Impact × Before versus After	4.513	3	1.504	1.627	0.182
	Error	532.713	576	0.925		
Orthocladiinae	Control versus Impact × Before versus After	0.036	1	0.036	0.036	0.851
	Spatial interaction × Before versus After	3.718	1	3.718	3.700	<b>0.050</b>
	Temporal interaction × Control versus Impact	0.594	1	0.594	0.591	0.839
	Stream × Control versus Impact × Before versus After	6.773	3	2.258	2.247	0.082
	Error	578.729	576	1.005		
<i>Zelandoperla</i>	Control versus Impact × Before versus After	1.901	1	1.901	2.234	0.136
	Spatial interaction × Before versus After	0.067	1	0.067	0.078	0.377
	Temporal interaction × Control versus Impact	0.380	1	0.380	0.447	0.504
	Stream × Control versus Impact × Before versus After	9.380	3	3.127	3.674	<b>0.012</b>
	Error	490.246	576	0.851		

Densities of the four most common and widespread taxa showed few statistically significant interaction terms that suggested an adverse effect of 1080 (Table 5). *Helicopsyche* densities were unaffected by the 1080 baits, whereas Orthocladiinae midge densities showed a significant effect only for the Spatial interaction × Before versus After, with an

increase at all sites following addition of 1080 (Fig. 5). Densities increased by 65% and 48% at the impact sites 100 m and 10 m downstream of the baits, respectively, which was greater than the observed increase in midge density at the control sites 10 m upstream of the baits (Fig. 5). *Deleatidium* densities showed a significant Temporal interaction



**Fig. 7** Densities of *Deleatidium* and *Zelandoperla* ( $\bar{x} \pm 1SE$ ,  $n = 40$ ) collected from individual streams from control (open bars) and impact (closed bars) sites, before and after 1080 (sodium fluoroacetate) bait placement.

× Control versus Impact effect (Table 5). Densities increased at all sites before and after addition of 1080 baits, except for the impact sites 4 days after the baits were added, when densities decreased by 18% (Fig. 6).

Patterns in *Deleatidium* and *Zelandoperla* densities differed between streams (Table 5). *Deleatidium* densities decreased slightly in Crooks and Small creeks at impact sites when compared with control sites, whereas in Dead Horse tributary, *Deleatidium* densities remained the same at impact sites before and after addition of 1080 baits (Fig. 7). Densities increased at both control and impact sites at Wheelbarrow creek. *Zelandoperla* densities changed little at both control and impact sites during the study at Crooks and Wheelbarrow creeks, whereas densities

increased slightly at impact sites at both Small creek and Dead Horse tributary after the addition of 1080 baits (Fig. 7). Densities of this taxa increased more in the control sites during the study.

## DISCUSSION

### Effects on native fish

We placed 80 Wanganui No. 7 baits in Wheelbarrow creek, and monitored fish mortality at a site c. 10 m below this. Use of 80 baits represented an “extreme” instance of contamination by 1080 baits over this short distance. Similarly high numbers of baits were placed in the other three streams, proportional to their discharge. If all the 1080 had leached from

the baits within 8 h, estimated 1080 concentrations would have ranged from 0.18 to 0.25  $\mu\text{g litre}^{-1}$  (Table 1), similar to what was observed. The highest 1080 concentration at Wheelbarrow and Wallaby creeks (0.26  $\mu\text{g litre}^{-1}$ ) was almost  $10 \times$  lower than the Ministry of Health's upper value of 2  $\mu\text{g litre}^{-1}$ , despite the very large quantities of baits placed in a small area in each stream. These results highlight the rapid rate at which 1080 leaches from submerged baits (see Suren 2006), and also support assertions by Eason et al. (1999) that 1080 leaching from baits is diluted to toxicologically insignificant amounts, even when sampled a short distance away from large numbers of baits submerged in small streams. We also found that 1080 concentrations were almost three times higher at sites 10 m below the baits than sites 100 m downstream. Although this downstream dilution effect may have reflected incomplete mixing of 1080 by 10 m, we consider it unlikely, as 1080 baits were placed at three places across each stream in relatively narrow sections, and stream turbulence was high as it flowed over the course substrate. More likely reasons for the lower 1080 concentrations at the 100 m site include groundwater inputs to the stream, and losses of 1080 from the stream from either bacterial breakdown or losses to groundwater.

Despite placing a large number of baits in each stream and detecting 1080 in water, no fish died after application of 1080 baits. The only mortality observed was that of 15 upland bullies and of an individual koaro as a result of a flood in Wheelbarrow creek that occurred before placement of 1080 baits. Lack of mortality of the three native fish species agrees with previous work showing that fish are tolerant to high 1080 concentrations ( $>370 \mu\text{g litre}^{-1}$ ; King & Penfound 1946; Batcheler 1978; Fagerstone et al. 1994). The highest of these experimental test concentrations was c. 3700 times higher than the highest concentration achieved in our study. To reach similar concentrations in small streams with a low discharge of 105 litres  $\text{s}^{-1}$  (as observed in Wheelbarrow creek when 1080 baits were added) and assuming complete dissolution of 1080 after only 4 h would require a total of 152 160 baits, or 973 kg of the 6.5 g Wanganui No. 7 baits. Such a scenario could only occur if a full hopper bucket carrying 1080 baits fell from a helicopter into a stream, all baits became submerged, and were left for 4 h. The chance of this occurring is minuscule. Moreover, standard operating conditions imposed on 1080 operations require any accidental spillages to be cleaned up as soon as possible, further reducing the likelihood of such a scenario.

### Effects on invertebrates

Stream discharges during the invertebrate study were lower than for the fish study, so fewer baits were used. Despite the use of fewer baits, 1080 concentrations were predicted to be higher than for the fish experiment, reflecting the reduced stream discharge and use of proportionally more baits. Notwithstanding these predictions, the water sampling protocol did not detect any 1080. Lack of detection reflected the rapid leaching rate of 1080, as the first samples were collected 24 h after addition of baits—well after all the 1080 had leached from the baits (Suren 2006). Predicted 1080 concentrations (assuming an 8 h leaching rate) would have ranged from 0.42 to 0.83  $\mu\text{g litre}^{-1}$ ; within the top 0.8% of 1080 concentrations recorded in water (Eason 2002). Given the close agreement between observed and predicted 1080 concentrations for the fish experiment, we believe these concentrations were attained in the invertebrate study. As such, the invertebrate study represented a worse-case scenario of 1080 contamination encountered within New Zealand streams.

Because 1080 is a toxin, densities of some invertebrate taxa were expected to decline following addition of baits as a result of mortality. This mortality would have reduced at least some of the calculated metrics at the impact sites, especially those closest to baits. Such reductions in metrics were not observed, and 1080 had little consistent demonstrable effect on metrics such as the MCI or %EPT. Although statistical differences were observed for some of the metrics, these differences were ecologically insignificant, and well within ranges of the metrics observed at control sites. For example, the small reduction in MCI scores at impact sites after addition of 1080 baits was insignificant (2) when considering the natural ranges of MCI scores observed at control sites (ranges from 88 to 186). The magnitude of the difference in the % EPT between Control versus Impact sites (3%) was also ecologically insignificant when compared to the range of %EPT at control sites (64–72%).

Examination of densities of individual taxa confirmed a lack of adverse effects of 1080 at the concentrations found in this experiment. In Crooks and Small creeks, *Deleatidium* densities decreased by c. 27% at impact sites following addition of 1080 baits. However, long-term studies of invertebrate communities in pristine headwater streams show that mean monthly densities of *Deleatidium* can vary up to 30% of the long-term (18-month) mean (Suren 1991, unpubl. data). We thus consider the

observed reduction in *Deleatidium* density at these two streams below 1080 baits more likely reflected natural variation, and not the presence of 1080 baits. Moreover, *Deleatidium* densities increased at sites below 1080 baits in Wheelbarrow creek, a result inconsistent with the effects of a toxin. Finally, the multivariate analysis detected no consistent changes to invertebrate community structure following application of 1080 baits in the study streams.

### Management implications

Given public concern about 1080 in the environment—especially when it falls into streams—the results and implications of this study are expected to be of interest to both advocates and opponents of the aerial application of 1080 baits. Consent conditions controlling aerial applications of 1080 usually implement buffer strips around larger rivers, presumably for protection of aquatic ecosystems and/or ensuring potential drinking water supplies remain uncontaminated. There is currently no national standard that we are aware of for imposing buffers around waterways during 1080 operations. Some regional councils require buffers around all waterways, others have buffers around all waterways >3 m wide, whereas other councils have no buffers around waterways that are not used for drinking water (Suren 2006). If one of the intents of buffer zones around streams is to minimise potential adverse effects of 1080 on the environment, then placing buffers on only streams >3 m wide will not prevent smaller headwater streams from being exposed to accidental contamination of 1080 baits.

The dilution capacity of a stream would be proportional to its discharge, so it could be argued that buffers should also be placed around smaller streams as they would have a lower capacity to dilute potential toxins. However, requiring buffers around small streams within an operational area would reduce the cost-effectiveness of aerial 1080 operations, as large areas of catchments would become no-drop zones, increasing the operational difficulty. Additionally, buffer zones around waterways may create refuge zones for the target species. Creation of such refuge zones may diminish the overall effectiveness of 1080 operations in specific areas.

Our experiments were performed in small streams <3 m wide, which could have been subject to potential contamination by 1080 baits during aerial operations. Despite placing large quantities of bait into these streams, we detected no adverse effects on three native fish species or natural invertebrate communities exposed to 1080 leaching from these

baits. If no adverse effects of 1080 were detected in small streams with a low dilution capacity, then there would be far less potential for adverse effects on biota in larger streams with a greater dilution capacity. Placing buffer zones around larger streams for the purpose of ecosystem protection therefore seems unjustified. Consent granting authorities should consequently consider removing requirements of buffer zones around large rivers. An exception to this recommendation may be when rivers are used for human drinking water or for stock supply, as some members of the public may still need assurance that all steps are being taken to prevent 1080 baits landing in these streams.

Finally, although our experiments were conducted over only a short-term period and failed to detect longer-term or sub-lethal impacts, our detailed water-monitoring programme during the fish experiment showed unequivocally that 1080 was rapidly lost from submerged baits within 12 h. This rapid loss of 1080 from submerged baits was reinforced by absence of detectable 1080 in water samples collected 24 h after addition of baits in the invertebrate experiment. Moreover, 1080 is broken down relatively quickly by bacteria present in water or associated with aquatic plants (Ogilvie et al. 1996). We consequently contend that no long-term effects of 1080 would be possible to fish or invertebrate communities as a result of minute quantities of 1080 leaching from submerged baits as the chemical would not be present in the environment for long enough.

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